Evaluating the costs and sampling adequacy of a vertebrate monitoring program

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ABSTRACT

Ecological monitoring is important for tracking trends in species and ecosystems over time and is the basis of conservation planning and government policy. Given there are increasing constraints on funding opportunities for conservation research there need to be simple approaches to assess the costs and effectiveness of surveys that highlight where methods can be refocussed to address changing management aims. In this study we use data from a vertebrate fauna monitoring program to assess the extent to which the sampling has been effective in recording the total estimated vertebrate species richness, identify which classes, families or functional groups of birds, mammals and reptiles have been under sampled by existing methods and which of the survey methods used were most cost effective. We compiled data collected over six surveys conducted over five years on a conservation reserve in northern Australia as a case study. We used rarefaction curves to examine the rate of species accumulation and sampling adequacy for 15 fauna functional groups representing bird, mammal and reptile taxa. We also compared the cost effectiveness using the relative dollar cost for six survey methods including both observational (active search, bird counts) and trapping (cage, box, funnel and pitfall traps) techniques. In our case study, despite six repeated surveys, with a total estimated cost in excess of $500 000, sampling for six of the 15 targeted fauna groups was insufficient. Multiple survey methods are required to sample taxa such as reptiles and small mammals. Costs per methods were approximately equal when comparing different techniques such as pitfall, cage, Elliott and funnel traps. This study demonstrates that it is straightforward to use simple metrics of survey success to guide, refine and improve monitoring programs.

Key words: biodiversity, monitoring, conservation, management; species accumulation curves, costs-effectiveness

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Introduction

Ecological monitoring is important for tracking trends in species and ecosystems over time and is the basis of conservation planning and government policy (Haughland *et al.* 2010; Thompson 2007). An essential component of successful monitoring is the use of appropriate methods that match a clear and defined goal for the survey (Eyre *et al.* 2011). For example, terrestrial vertebrate monitoring can be targeted and specialised for a single taxon or multi-taxa monitoring to examine change in response to management or other pressures relative to each other over time (Eyre *et al.* 2011). Regardless of the monitoring goal, data must be collected in an effective manner, analysed appropriately and surveys should be periodically evaluated to assess their relevance to specific program aims, their efficiency and where changes or refocussing can be made over time in line with new research hypotheses, management goals or public policy priorities (Pullin *et al.* 2004; Tulloch *et al.* 2011). Measures such as sampling adequacy and cost-effectiveness are an important component of this (Ribeiro *et al.* 2008), though in reality calibrated indices of population and occupancy and population estimates, that are not commonly used in Australian studies, should be more widely and consistently applied (Wintle *et al.* 2005).

Multiple methods are used for detecting different species and taxa (see reviews in Garden *et al.* 2007; Sutherland 1996) with certain methods clearly more effective for targeting particular species over other ones (Garden *et al.* 2007). In many cases the survey methods chosen can be influenced the pre-existing sampling regime and logistics rather than strictly on their effectiveness which may vary with geography or climate. However, methods can be compared based on their adequacy to sample different taxa and cost effectiveness (Garden *et al.* 2007; Hughey *et al.* 2003; Ribeiro *et al.* 2008), which describes monitoring costs relative to a defined output (Hughey *et al.* 2003). Accounting for the financial costs in context of the ecological outcomes of a project is becoming an increasing focus of current government biodiversity policy and investment (Morrison *et al.* 2010).

In this study we use terrestrial fauna survey data collected during six surveys conducted over a five year surveillance monitoring program (Kutt *et al.* 2012) to assess survey effectiveness using sampling adequacy (in this case species accumulation over time) and cost metrics. Surveys commenced following the removal of cattle from a private conservation reserve, with the specific aim to document change in the terrestrial vertebrate assemblage over time. The patterns of change and recovery in species composition and abundance over time have previously been examined (Kutt *et al.* 2012). In this study

we use the data to examine some targeted questions regarding monitoring namely: (i) the extent to which the sampling program has been effective in recording the total estimated vertebrate species richness; (ii) which classes, families or functional groups of birds, mammals and reptiles have been under sampled by existing methods; and (iii) which of the survey methods used were most cost effective. Though our case study is specific to a region in north-eastern Australia, these questions are universal to all global conservation programs (Gardner *et al.* 2008).

Materials and methods

Study area

The study was conducted on Brooklyn Wildlife Sanctuary near Mt Carbine, north east Queensland (145° 9'E, 16°31'S) in the Einasleigh Uplands bioregion. Brooklyn station is a 60,000 ha property owned and managed since 2004 by a non-government, not-for profit conservation organisation, the Australian Wildlife Conservancy (http://www.awc.org.au/). The vegetation is predominantly mixed open *Eucalyptus*, *Corymbia* and *Melaleuca* tropical savanna woodlands. The region experiences strongly seasonal rainfall, varying annually from 350 mm to >1500 mm (Bateman *et al.* 2010).

Monitoring at Brooklyn station was undertaken to determine the responses of vertebrate fauna to the cessation of cattle grazing. Initially, broad surveillance surveys were conducted (Kutt *et al.* 2012), along with targeted altitudinal surveys from rainforest to savanna habitats (Bateman *et al.* 2010; Kutt *et al.* 2011). The data was intended to inform property management, especially as it relates to the use of fire. Fifty sites were selected within the property (40 across the savanna vegetation, 10 on the altitudinal gradient), stratified by broad vegetation type. Full details are provided in Kutt *et al.* (2012).

Survey methods

Vertebrate fauna surveys were conducted six times (May 2006, November 2006, April 2007, November 2007, April 2010, November 2010), sampling both the end of wet and dry seasons in each year. Fauna sampling at each site occurred within a standardised 1 ha (Kutt and Fisher 2011). Nested in the 1 ha quadrat is a 50 x 50 m trap array comprising twenty box traps (Elliott Scientific Equipment, Upwey, Victoria, hereafter termed Elliott traps) for sampling small mammals, two wire cage traps (for sampling medium-sized mammals and four pitfall traps (60 cm deep and 30 cm diameter, 10 m apart and arranged in a 'T' configuration with 20 m and 10 m of drift fence), for sampling small mammals and reptiles and six funnel traps for reptiles. Trapping was supplemented by timed observational searches: three diurnal and two nocturnal searches each of 20 person-minutes duration and 8 five-minute diurnal bird counts were conducted over a four day period (Perry *et al.* 2012).

Analysis

An estimated maximum number of species was calculated for each family or group, as specified by the asymptote of rarefaction curve. We then calculated the number of individuals and species required to be sampled to reach 90% of the predicted maximum; an acceptable lower confidence limit for sampling (Moreno and Halffter 2000; Thompson *et al.* 2007).

We examined sampling confidence (90% of the total predicted number of species, based on Michael-Menis non-linear regression curves used to determine the rarefaction asymptote – see further description below) based on class (birds, mammals, reptiles); family (reptiles only); and by life history attributes (birds and mammals). Birds were grouped as nocturnal (N), ground dwelling (GD) or other diurnal (O) species according to foraging and life history data (Kutt and Martin 2010; Tassicker *et al.* 2006). Mammals were grouped by their predominant activity period (nocturnal or diurnal) (Menkhorst and Knight 2004), body weight (> or < 5 kg) (Johnson and Isaac 2009) and dominant activity strata (arboreal versus ground-dwelling) (Van Dyck and Strahan 2008). This resulted in three mammal groups for analysis: diurnal large (DL), nocturnal small arboreal (NA) and nocturnal small ground dwelling (NG).

Rarefaction curves were plotted for each group of vertebrates using EstimateS Version 8.2 (Colwell 2009). The number of species observed was plotted against the predicted number of individuals using the Mao Tau estimate. This method provides an estimation of the number of species per number of individuals sampled based on 1000 random iterations (Nichols *et al.* 2011). Individuals rather than samples was chosen as the unit of comparison as it provides a measure of species richness rather than density thus allowing species accumulation curves to be directly comparable across groups and taxa (Gotelli and Colwell 2001). Non-linear regression curves used to determine asymptotes were fitted using the model fitting program zunzun (http://zunzun.com/) (Phillips 2010), using the average number of species observed (in this case the Mao Tau estimate as generated by EstimateS). Prior to the model fitting, several non-linear models were compared to determine the method of best fit (Beta- P, Scaled Power, Weibull and Michaelis-Menten) and assessed using the criteria defined by Toti et al. (2000). The Michaelis-Menten model (Clench 1979) provided the best fit across all groups and is commonly used to predict species richness in heterogeneous areas where rare species are present (Toti *et al.* 2000). To test the strength of our predictions we also compared the total number of species estimated using rarefaction to the possible list of species which could occur within the Brooklyn area using known species distributions and expert opinion

Cost analysis

We defined cost effectiveness as the relative dollar cost of each survey method per number of species recorded. The total cost for each technique was calculated per site, including initial set up costs and costs for subsequent surveys. Costs were categorised by labour, equipment (initial and on-going) and travel, with all values reported in Australian dollars. Labour rates were calculated based on an average salary of a research scientist ($50/hour). Estimated labour time included preparation of equipment, travel to and from sites, set up time, checking traps, conducting bird surveys and active searches, rebaiting and repairing traps and final pack up. Travel time was standardised at 30 minutes per site, an estimate between the shortest and

longest required travel time (10 - 70 min). A minimum of 15 minutes per site was allocated to check traps.

Equipment expenses comprised initial set up costs including; purchasing traps and equipment and excavator hire for installing pitfall buckets. Ongoing costs included consumables (batteries, bait) and depreciation of the equipment (active search tools and binoculars). Consumable costs were based on supermarket purchase prices and scaled to the quantity required for each technique. Equipment usage costs were calculated as the initial purchase cost divided by average number of uses of the equipment over its lifetime (estimated at number of searches by average number of annual surveys conducted per year over a 10 year lifespan, e.g. hand rakes = $0.08 / unit search calculated by cost ($15) / (3 active searches x 6 surveys x 10 years).

Travel costs were calculated on an average distance of 30 km per site, multiplied by the number of visits required. In practice, travel and labour costs would be minimised by checking multiple traps at a site on the same visit; for example, undertaking a bird count and checking pitfall traps. However, to allow comparison, each cost was calculated independently, hence estimated costs represent the maximum possible cost incurred. For each method the total cost was calculated for the initial installation and repeat surveys. We used metrics outlined by Garden *et al.* (2007) to calculate the cost effectiveness of each trapping method for each taxa or group. Calculations were based on the average number of species detected per site divided by the average survey method cost per site. The total number of species and number of unique species (those only captured by a single method) were also calculated.

Results

Sampling confidence levels

Across the five years, over 21 000 individual vertebrate records were collected from 216 species comprising 128 bird, 62 reptile and 26 mammal species. Within each class, the group or family with the highest number of individuals also had the highest species richness, specifically the groups of other diurnal birds (O), Scincidae for reptiles and nocturnal small ground dwelling (NG) mammals (Figure 1). We could

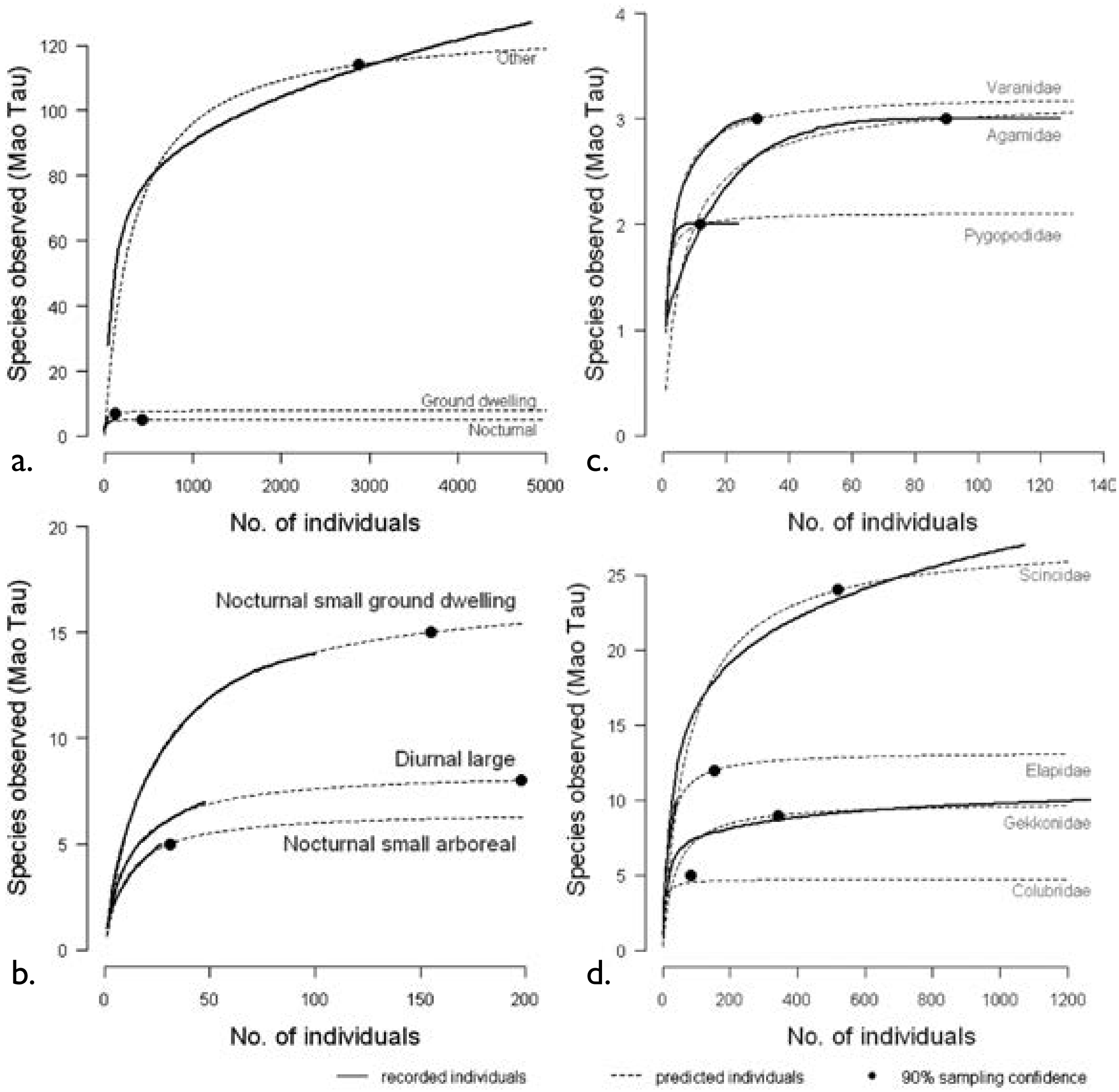

Figure 1. Sample-based rarefaction for birds (a), mammal (b) and reptile (c and d) groups. Curves were plotted for each group of vertebrates using EstimateS and the number of species observed was plotted against the predicted number of individuals using the Mao Tau estimate.

not calculate species accumulation curves with any certainty for Boidae and Typhlopidae (reptiles) due to low number of individuals recorded (n = 6 and 4, respectively).

Confidence in sampling varied between groups and classes (Table 1). For example, sampling for all reptiles combined met the 90% confidence limit; however three of the nine reptile families (Colubridae, Elapidae and Typhlopidae) remained under sampled at the 90% sampling confidence threshold. Similarly, while all birds combined reached 90% sampling confidence after the first year, ground dwelling (GD) birds remained under sampled. For mammals, only the NA group reached the 90% confidence level after three years of surveys.

Of the six groups which remained under sampled (GD birds, Colubridae, Elapidae, DL and NG mammals), less than two additional species were needed to be captured to reach the 90% confidence level. However, due to the variation in total species richness within each group, the proportional number of species ranged from 7% (NG) to 20% (Colubridae). Similarly the number of additional individuals required to be recorded to meet the 90% confidence threshold ranged from 83 (Colubridae) to 196 (DL mammals).

Comparison of the number of species predicted by species accumulation curves was similar to those predicted based on species distribution data (Table 1). On average, regression extrapolated totals were 2.5 species fewer than those as predicted using distribution maps and expert opinion. Nocturnal species (birds and small mammals) were significantly underestimated (6 species).

Both Agamidae and Pygopodidae reached an asymptote with few individuals; however the number of species within these groups is limited. Conversely groups with many species, such as Scincidae and other diurnal bird species (group O) showed potential for more species to be recorded, despite high sample numbers. By contrast, families such as Varanidae and Gekkonidae which have few species, but where many individuals were recorded was well above the 90% confidence level.

Costs of survey methods

The cost of survey varied widely across the six methods (Figure 2) with total costs over the five years highest for bird counts ($266,100) and lowest for active searching ($111,460). Initial set up costs (that includes capital costs) were Elliott traps ($1,374), bird counts ($1,109), funnel traps ($876), cage traps ($795), pitfall traps ($711), and active searches ($464). For repeat surveys, costs remained relatively similar for bird counts and active searches, but decreased for Elliott traps ($614), cage traps ($593), pitfall traps ($570) and funnel traps ($515). This would be expected, given the high

Table 1. Total number of individuals and species of birds, mammals and reptiles recorded during the survey, predicted number of individuals and species required to reach sampling confidence†, proportional number of species (%) required to reach sampling confidence; year of survey in which sampling reached 90% sampling adequacy; and number of species predicted to occur in the region.

Group	Individuals recorded (n)	Species recorded (n)	Species predicted (n)	Individuals predicted (n)	Proportion of species required (%)	Year	Species predicted in region (n)
Birds	12298	128	124				
Nocturnal	150	5	5			3	11
Ground Dwelling	47	6	7	125	17	Na	8
Other Diurnal	12101	127	114			1	na
Reptiles	8483	62	59				
Agamidae	446	3	3			1	5
Boidae ‡	6	1	1				5
Colubridae	19	4	5	83	25	na	4
Elapidae	41	10	12	153	20	na	18
Gekkonidae	3290	10	9			1	11
Pygopodidae	27	2	2			2	3
Scincidae	4598	27	24			1	30
Typhlopidae ‡	4	2	3				4
Varanidae	52	3	3			2	4
Mammals	258	26	29				
Diurnal Large	57	7	8	196	14	na	10
Nocturnal Small Ground Dwelling	170	14	15	155	7	na	23
Nocturnal Small Arboreal	31	5	5			3	11

† Sampling confidence defined as 90% of the total predicted number of species, based on Michael-Menis regression curve (y = ax / (b + x)

‡ indicates the number of individuals recorded was too low to calculate species accumulation curves

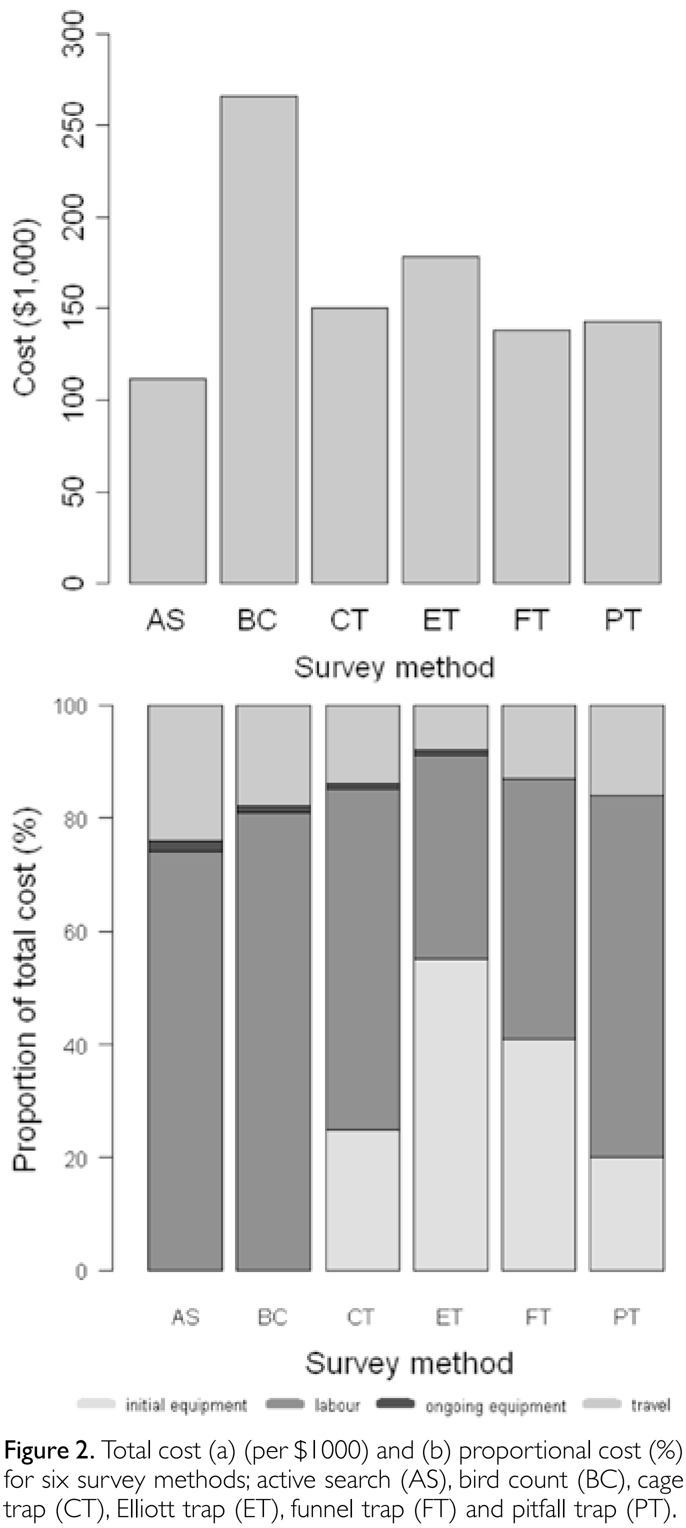

Figure 2. Total cost (a) (per $1000) and (b) proportional cost (%) for six survey methods; active search (AS), bird count (BC), cage trap (CT), Elliott trap (ET), funnel trap (FT) and pitfall trap (PT).

proportion of cost attributed to initial equipment purchase and installation. The majority of costs were due to labour, followed by travel, with only a small proportion of cost associated with ongoing equipment and maintenance (<1%).

Cost effectiveness

To determine the outputs per cost, we calculated the average number of species recorded per dollar spent for each method (Garden *et al.* 2007). Methods which recorded a high number of species per dollar were considered more cost effective. Overall the most cost efficient survey method depended on the class, family or group (Table 2). Not surprisingly, bird counts were the most cost efficient method of recording the majority of bird species (2.7 species per $100). For the bird groups N and GD active searches were most cost effective (0.31 and 0.22 species per $100). Active searching was the most cost effective method for detecting all families of reptiles except Typhlopidae (which was only recorded from pitfall traps) (Table 2). Despite this, four of the five trapping methods detected unique species not detected during active searches. For example, six of the ten elapid species were captured only in funnel traps (Table 2). Active searching was the most cost efficient method for all groups of mammals combined (0.24-0.22 species per $100). DL and NA mammals were best detected by active searching, while some NG mammal species were only recorded by pitfall or Elliott traps.

Discussion

Monitoring programs that evolve in response to research findings can provide more rigorous and cost effective data for conservation management (Nichols and Williams 2006). Typical approaches to evaluating survey data can include examination of statistical power (Steidl *et al.* 1997), but less frequent is evaluation of the survey effort adequacy and cost effectiveness once monitoring has commenced. Critical review of methods and results over time can highlight where possible changes can be made to refocus the surveys and improve the management outcomes (Colwell *et al.* 2004). Our case study demonstrates how a simple approach can be used to assess the effectiveness of survey methods whereby accumulated data can be used to identify data gaps, assess the adequacy and specificity of current methods and enable modifications of the program to improve overall effectiveness and efficiency (Lindenmayer and Likens 2010).

Wildlife surveys are an intensive and often costly way of monitoring species within an area of interest (Margules and Austin 1991). In our case study, despite six repeated surveys, with a total estimated cost in excess of $500 000, sampling for six of the 15 targeted fauna groups was insufficient. Under detected groups were typically cryptic (e.g. GD birds like button-quails, *Turnix* spp., large elapid snakes), subterranean (e.g. Typhlopidae) or trap shy (e.g. large elapids snakes). These taxa, which include threatened species, species at low abundances and species by nature difficult to detect in any landscapes (Schutz and Driscoll 2008), highlight gaps in current data collation at Brooklyn Station. In contrast the detection of groups such as Scincidae, Agamidae, and Gekkonidae and the majority of diurnal birds (excluding nocturnal and ground dwelling) species was high. This disparity suggests that resources can be redirected in future surveys to target those in underrepresented groups, for example, if the future objective is to provide data on management of mammals, which are a particular conservation concern in northern Australia (Kutt 2012) and a flagship management species for private conservation reserves (Kutt *et al.* 2012).

Table 2. Summary of bird, mammal and reptile trapping success as total and unique (in parentheses) number of species and cost effectiveness as number of species per $100 for six trapping methods; active search (AS), bird count (BC), cage trap (CT), Elliott trap (ET), funnel trap (FT) and pitfall trap (PT).

Group	Total and unique number of species recorded						Average number of species / $100 cost					
	AS	BC	CT	ET	FT	PT	AS	BC	CT	ET	FT	PT
Birds												
Nocturnal	5 (3)	2					0.31	0.1				
Ground Dwelling	2 (1)	5 (4)					0.22	0.1				
Other Diurnal	22 (1)	126 (104)	2	1			0.24	2.7	0.15	0.12		
Reptiles												
Agamidae	3			1	2	3	0.22			0.12	0.16	0.18
Boidae	1 (1)						0.21					
Colubridae	3 (2)				2 (1)		0.23				0.16	
Elapidae	4				10 (6)		0.22				0.19	
Gekkonidae	10 (2)			2	7	8	0.68			0.12	0.21	0.3
Pygopodidae	2				1	2	0.24				0.16	0.16
Scincidae	21 (2)		2 (1)	2	17	22 (4)	0.8		0.23	0.12	0.36	0.53
Typhlopidae						2 (2)						0.17
Varanidae	3		1	2	2	2	0.22		0.15	0.12	0.16	0.16
Mammals												
Diurnal Large	7 (7)						0.24					
Nocturnal Small Ground Dwelling	7 (1)		4	10 (2)	1	7 (1)	0.24		0.11	0.17	0.16	0.16
Nocturnal Small Arboreal	5 (4)		1				0.22		0.15			

While capture success is commonly influenced by environment, landscape and timing factors (Cunningham *et al.* 2005; MacKenzie *et al.* 2003; Thomas *et al.* 2010; Vine *et al.* 2009), we found the success of a method was also influenced by life history traits and body size of the target species. Pitfall and funnel trapping methods detected many unique species across multiple taxa whilst active searching was most effective across all groups, specifically for large bodied taxa and rare or cryptic species. Our results support the recommendations of other authors that multiple methods be used to sample a range of reptiles (ie. skinks and snakes, Ribeiro *et al.* 2008) and small mammals (Naxara and Pardini 2006). Low detection rates for mammals suggest that significantly more effort is required for to reach the 90% confidence threshold, and without greater detection, comparison between species, sites and habitats is more difficult. GD mammals were best monitored by using a combination of traps including Elliott, pitfall, and cage traps which supports previous work in this field (Catling *et al.* 1997; Cunningham *et al.* 2005; Ribeiro *et al.* 2008). Bird counts and active searching are the most cost-effective survey methods, although both require a high degree of expertise to minimise observer bias (Lindenmayer *et al.* 2009). Novel techniques such as funnel traps are cost effective for capturing snakes, which are often under sampled in general surveys (Woinarski *et al.* 2000).

Variation in cost per method was due to the person-hours required or the initial cost of equipment, which was most apparent when comparing observational and live trapping techniques. The high contribution of labour costs (60%) recorded within this study, fell within the range (38 – 71%) previously reported for multi-taxa vertebrate surveys (Gardner *et al.* 2008). Labour costs increased with frequency of sampling, for example the cost of bird counts, which required eight visits per site, were significantly higher than active searches which required only two. However, the success of observational methods is largely due to observer experience, and whilst this can be captured in some respect by cost of person per hours (Gardner *et al.* 2008), the availability of trained and experienced observers capable of conducting reliable surveys is often a limiting factor (Ribeiro *et al.* 2008). Volunteers can offset these costs, but can lead to unreliable data (Newman *et al.* 2003). Direct comparisons between studies are difficult due to variation in calculated costs (Garden *et al.* 2007) and variation of survey locations, substrate and logistics. Alternatively Ribeiro *et al.* (2008) excluded travel from cost estimates on the assumption that all methods would incur equal costs. Despite this complication, costs were similar when comparing pitfall, cage, Elliott and funnel traps (this study); pitfall and funnel traps (Ribeiro *et al.* 2008) or cage, Elliott and pitfall traps (Garden *et al.* 2007).

While the use of species accumulation curves provides a conceptually attractive application for critically reviewing monitoring objectives and success, there are some limitations. Gotelli and Colwell (2001) discourage the use of species accumulation curves for extrapolation on the basis that accurate asymptotes are unable to be adequately predicted in nature. Insufficient samples or data which are seasonally or temporally skewed will result in incorrect estimates (Soberon 2009), as indicated by the families Boidae and Typhlopidae within this study. Similarly populations with rare or undetected species, those in

heterogeneous environments or those where individuals will migrate are likely to underestimate total species numbers (Preston 1964). These trends were mirrored in this study through lower total species richness estimates predicted by non-linear regression as compared to known species potential distributions. Clearly, stable species richness levels are rarely achieved in a natural system and often the aim is to detect relative changes due to management intervention (Gardner *et al.* 2008; Ribeiro *et al.* 2008; Thompson *et al.* 2003). In spite of these limitations, we believe that species accumulation curves provide a simple method to estimate adequacy of sampling and this has been suggested previously for invertebrate surveys (Oliver and Beattie 1996) and biodiversity surveys in general (Thompson *et al.* 2007).

Conclusions

In this study we have demonstrated that for vertebrate fauna surveys aimed at investigating the impacts of land management decisions at Brooklyn Wildlife Sanctuary, it is quite easy to use simple evaluation tools to critically examine which methods are successful and which require modification for future sampling. Importantly this includes optimising return on effort and expenditure. Our case study indicates after five years of standardised surveys, a number of taxa and functional groups stand out as being under-sampled using the current methods. A prime focus of the Australian Wildlife Conservancy is the conservation and protection of Australian wildlife with mammals a flagship group in many instances, and the application of property wide management programs to enhance and promote mammal population stability and recovery from threatening processes (Kutt *et al.* 2012). In this study we found using current methods, nocturnal small ground and small arboreal mammals were under sampled and future monitoring should be refocussed to place more effort into these taxa. However, as long term ecological monitoring is a significant activity for identifying critical changes in fauna due to anthropogenic or other broad environmental effects (Perry *et al.* 2011), continuation of broad scale surveillance monitoring that maintains the standardised sampling regime is also recommended (Eyre *et al.* 2011). Though many of our results might seem intuitive, it demonstrates the simplicity and value of the task of reviewing the costs and effectiveness of surveys over time to refine monitoring. These sorts of analyses ensure that data provides meaningful information for targeting future investment, and application to future environmental action undertaken by conservation managers and government agencies (Morrison *et al.* 2010).

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